EMISSIONS FROM SIMULATED OPEN BURNING OF DEPLOYED US MILITARY WASTE

THESIS

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AFIT/GES/ENV/12-M05

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Abstract

U.S. military forces have relied primarily on open burning as an expedient method of volume reduction and treatment of solid waste during the conflicts in Afghanistan and Iraq. This study is the first effort to characterize a broad range of pollutants and their emission factors during the burning of military waste and the effects that recycling efforts, namely removing plastics, might have on emissions. Piles of simulated military waste were constructed, burned, and emissions sampled at the U.S. Environmental Protection Agency (EPA) Open Burn Testing Facility (OBTF), Research Triangle Park, NC. Three tests contained polyethylene terephthalate (PET #1 or PET) plastic water bottles and four did not. Emission factors for polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), particulate matter (PM_{10}, $PM_{2.5}$), polychlorinated and polybrominated dioxins/furans (PCDD/F and PBDD/F), and criteria pollutants were determined and are contained within. The average PCDD/F emission factors were 270 ng-toxic equivalency (TEQ) per kg carbon burned (ng-TEQ/kg Cb), ranging from 35-780 ng-TEQ/kg Cb. Limited testing suggests that targeted removal of plastic water bottles has no apparent effect on reducing pollutants and may even promote increased emissions.

To my wife and children, for sacrificing the warm beaches of Okinawa and the Carolinas.

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Brian D. Woodall

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EMISSIONS FROM SIMULATED OPEN BURNING OF DEPLOYED US MILITARY WASTE

I. Introduction

General Issue

Military forces throughout history have suffered higher percentages of deaths from illnesses, due largely to poor sanitation practices, rather than from enemy actions. This was recognized as early as 1861 when the United States created the U.S. Sanitation Commission (USSC) to promote clean and healthy living conditions within Union Army camps to reduce these losses. Since these early efforts, U.S. military sanitation practices have greatly reduced personnel losses due to poor sanitation, achieving lower losses from accidents, disease, and infections than enemy actions for the first time ever during World War II (Leland, 2010). Of these sanitation practices, the disposal of solid waste is a major factor.

Military operations produce large quantities of solid waste, which quickly accumulates as units begin to establish more permanent positions. This accumulation is due primarily to the lack of sufficient and safe off-base disposal methods in the deployed environment and the lack of treatment technology, such as incinerators. In order to avoid unsanitary accumulation of waste, units rely heavily on the use of open burn pits as an expedient method of volume reduction and treatment. An open burn pit is an excavated pit where solid waste is consolidated and burned. Open burn pits allow units to reduce the threat from disease and the volume of waste while remaining in a secure compound. This has been a long practiced method of waste treatment during military operations.

The permanent positions discussed earlier are, today, known as Forward Operating Bases (FOBs). FOBs serve as the primary means for providing a secure location for bivouacking troops during military operations. In fact, nearly all military personnel that have served in Iraq or Afghanistan have spent some time living on a FOB. Also, FOBs can serve a wide range of purposes from temporary outpost that move with combat units and the Forward Edge of the Battle Area (FEBA), to major logistical hubs that provide security for thousands of personnel conducting many services in support of combat and stability operations. These services include all forms of logistical and aviation support. Additionally, the large logistical FOBs can house thousands of personnel permanently and facilitate the transition of entire units between Areas of Responsibility (AORs), which can temporarily increase the FOB's population by thousands. The diversity of services performed and the amount of personnel living on these FOBs create a waste stream that is potentially different in both composition and amount than that of municipal waste streams.

Several factors influence the unique composition of military waste. First, nearly all military supplies and equipment are packaged and shipped into theater, which increases the percent of shipping materials found in the waste stream. Also, due to the limited water resources and treatment facilities, most consumable water supplies are shipped to FOBs in polyethylene terephthalate (PET #1 or PET) plastic water bottles for nearly all daily water requirements. These water bottles remain in the waste stream since recycling efforts are largely nonexistent, due to lacking infrastructure and local demand. Meal Ready-to-Eat (MRE) waste, which is a packaged daily ration for soldiers, may be found in large quantities in the waste stream. Furthermore, heavy canvas materials in the

form of tents, uniforms, cots, and weather proofing are commonly destroyed and discarded during military operations. Overall, the waste stream consists of large amounts of plastics, styrofoam, electronics, and other items that may produce harmful emissions when burned (Army Institute of Public Health, 2010; Hardt, 2011).

As previously stated, the amount of waste produced at deployed locations is also of concern. There have been many studies of the amount of solid waste produced by personnel during military operations with widely ranging estimates. A 2009 study analyzing the waste generation streams in Iraq and Afghanistan concluded the solid waste generation rate to be approximately 10 lbs/person/day (CH2MHILL, 2009). Also, during fiscal year 2008, average troop levels in Iraq and Afghanistan peaked at approximately 187,900 personnel (Belasco, 2009). Therefore, during peak troop levels, U.S. military operations were producing more than 900 tons of solid waste per day. The vast majority of this waste was disposed of in open burn pits. For instance, records show that civilian contractors on Joint Base Balad (JBB), Iraq, were burning nearly 200 tons per day of solid waste during peak troop levels (Armed Forces Health Surveillance Center: Naval Health Research Center & US Army Public Health Command, 2010). Due to the amount of waste disposed and limited stand-off distance due personnel security measures, personnel deployed to Iraq and Afghanistan may have been exposed to harmful emissions in excess of Military Exposure Guidelines (MEGs).

In a cohort study conducted from 2001 to 2006, military personnel deployed to these regions had a higher rate of newly reported respiratory symptoms than non-deployers, which was 14% vs. 10%, respectively (Smith et al., 2009). Also, rates of respiratory symptoms necessitating spirometry among personnel returning from Iraq and

Afghanistan as opposed to non-deployers were found to be 14.5% and 1.8%, respectively (Szema et al., 2011). The latter study uses the term Iraq/Afghanistan War Lung Injury (IAW-LI) to describe a host of pulmonary complaints from personnel returning from these geographic regions during the ongoing wars. This study also implicates exposure to open burn pit emissions as a possible etiologic mechanism for new-onset IAW-LI (Szema et al., 2011).

In response to the growing concern for service-members exposure to harmful contaminants from open burn pits and increases in respiratory illnesses in returning veterans, Congress has directed that studies be conducted to determine the long term health effects and that stricter regulations governing the operation of burn pits be imposed. In 2009, Congress signed into law Section 317, "Prohibition on Disposing of Waste in Open-Air Burn Pits," of the 2009 National Defense Authorization Act, H.R. 2647. This law requires that combatant commands are required to determine that there are no alternative disposal methods available and obtain permission from the Secretary of Defense in order to establish and operate open air burn pits, with review and permission renewal required every 180 days (Trimble, 2010). Additionally, U.S. Central Command (CENTCOM), the geographic combatant command whose AOR includes Iraq and Afghanistan, prohibited the burning of hazardous materials such as tires, treated lumber, batteries, and plastics in open burn pits (Trimble, 2010). Also, organizations such as the Government Accountability Office (GAO) have investigated the Department of Defense's (DOD's) adherence to guidance regarding burn pit operations to ensure compliance (Trimble, 2010).

In support of these actions, health services components in all military branches, with the Navy representing the Marine Corps, began research efforts which included improved air sampling techniques, retrospective exposure modeling, air curtain impacts, and more. Also, the Armed Forces Health Surveillance Center (AFHSC) and the Naval Health Research Center (NHRC) were tasked to conduct expedient epidemiologic studies using readily available data to determine any associations between exposure to burn pit smoke and illness (Armed Forces Health Surveillance Center: Naval Health Research Center & US Army Public Health Command, 2010). Additionally, the Department of Veterans Affairs (VA) commissioned the Institute of Medicine (IOM) to assess the long-term health effects from exposure to burn pits in Afghanistan and Iraq, and their findings were released in 2011 (Tollerud et al., 2011). All of these efforts have expressed the need to determine emission factors from open burning of solid waste at FOBs in order to improve knowledge of emissions.

A review of the literature revealed that concerns about emissions from open burning are not limited to the military. As regulations and treatment technologies have greatly reduced emissions from industrial and incinerator sources, open burning has been identified as a major global source of persistent organic pollutants (POPs), which are harmful to both humans and the environment. Exposure to POPs can lead to serious health effects including certain cancers, birth defects, dysfunctional immune and reproductive systems, greater susceptibility to disease and even diminished intelligence (Interim Secretariat of the Stockholm Convention, 2005).

The Stockholm Convention, which took effect in 2004, was established by the United Nations Environmental Program (UNEP) to eliminate production and use of POPs

and, where feasible, their release into the environment (Interim Secretariat of the Stockholm Convention, 2005; United Nations Environment Programme (UNEP) Chemicals, 2005). In accordance with the Stockholm Convention, many research efforts have been undertaken to identify and inventory sources and emissions of these POPs for the purpose of prioritizing and implementing emission controls. In 2004, Lemieux et al. published a comprehensive review of the literature related to emissions from open burning. The review showed that 125 studies had been conducted to characterize emissions from the open burning of various fuel sources, with the majority focusing on biomass and crude oil (Lemiux et al., 2004). However, recent research efforts have focused on anthropogenic fuel sources such as landfill and domestic waste burning due to their exceedingly high observed POP emissions.

One category of POPs, which has been the focus of much attention and is found largely in emissions from anthropogenic fuel sources, is polychlorinated dibenzodioxins and polychlorinated dibenzofurans (PCDD/Fs), known simply as “dioxins” and “furans.” These pollutants are highly persistent, bioaccumulating in the environment, and ending up in the food chain (Gullett et al., 2000). The more toxic forms have been labeled as “known human carcinogens” and have been linked to tumor formation, learning deficits, endometriosis, and immune-suppression (Gullett et al., 2000). Much of the published literature in regards to open burning emission is focused on these types of pollutants.

PCDD/Fs are each composed of eight homologue groups, distinguished by the number of chlorine (Cl) atoms attached to their ring structures. The eight homologue groups have a total of 210 congeners, which have varying numbers of structural isomers. Of these 210 congeners, 17 are considered toxic to varying degrees based on their

biological activity (Gullett et al., 2000). The World Health Organization (WHO) has assigned a toxic equivalency (TEQ) to each of these congeners in relation to 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD), which is the most toxic, for ease of comparison (Van den Berg et al., 2006). The formation of these pollutants during the combustion process is not well understood, but waste composition, orientation, and burn conditions are thought to be correlated (Gullett et al., 2000).

Research efforts to characterize the emissions from open burning of landfills have observed high concentrations of PCDD/Fs with large variances (Barakat, 2003; Gullett et al., 2010; Zhang et al., 2011). Additionally, laboratory simulations of the burning of domestic waste have had similar results (Gullett et al., 2001; Hedman et al., 2005; Lemieux et al., 2003; Lemieux et al., 2000). These simulated studies have varied the fuel composition by altering the content of polyvinyl chloride (PVC), a source of Cl, Cu, and other factors such as moisture to determine their contribution to the formation of PCDD/Fs (Lemieux et al., 2003). Additionally, Lemieux et al. compared the emission factors for the burning of domestic waste for households that did not recycle with households that avidly recycle, by removing bulk paper, glass, and plastics (Lemieux et al., 2000). It was determined that avid recyclers produced higher emissions of PCDD/Fs than those that did not recycle (Lemieux et al., 2000).

Ultimately, the literature indicates that PCDD/F emissions cannot be determined from the fuel source composition and that there have been no studies of open burning with waste similar to that found on FOBs. Additionally, the emission studies conducted thus far have had a limited scope of pollutants that have been investigated per effort.

Problem Statement

There are many published journal articles which characterize emission factors for various waste compositions and conditions. However, these emission factors have not focused on burning of deployed military waste and have been limited on the number of pollutants. It is hypothesized that the unique composition of this waste stream may produce significantly different emission factors than those observed in municipal waste burning and that emission factors may be altered with the enactment of recycling efforts.

Research Objectives

The purpose of this thesis is to determine emission factors for the open burning of simulated deployed military waste, for the broadest range of health-related pollutants investigated to date, and publish these results for their inclusion in future research efforts. Additionally, this thesis will determine the effects of removing plastics on emission factors. All emission factors will be calculated and published in units of compound mass per mass of carbon burned, which will allow the estimation of total mass of each compound produced.

The following is a list of specific questions used to guide this research:

1) What are the emission factors from open burning of simulated deployed U.S. military waste?
2) What are the effects of polyethylene terephthalate plastic (PETE or PET#1) recycling efforts on these emission factors?
    - 2.1: Is recycling beneficial given the additional resource requirements?

Scope and Approach

All research was conducted at the U.S. Environmental Protection Agency's (U.S. EPA) Open Burn Testing Facility (OBTF) located in Research Triangle Park, North Carolina. Representative military waste was constructed, per tri-service burn pit committee's estimations, and burned in triplicate for each condition, with and without plastics. Health-related compounds which included polychlorinated and polybrominated dibenzodioxins/dibenzofurans (PXDD/PXDF), polycyclic aromatic hydrocarbons (PAHs), aromatics, volatile organic compounds (VOCs), particulate matter 10µg or less (PM_{10}), $PM_{2.5}$µg or less ($PM_{2.5}$), and criteria pollutants such as; carbon monoxide (CO), carbon dioxide (CO_2), sulfur dioxide (SO_2), various forms of nitric oxides (NOx), and oxygen (O_2), were sampled during this experiment. An Analysis of Variance (ANOVA) was conducted for each compound to determine if there were any statistical differences between samples that contained plastics and those that did not. Once these emission factors were calculated and compared to related emission studies, they were submitted in a scholarly journal to *Environmental Science and Technology* for use in future research and regulation efforts.

Significance

All previously mentioned research efforts have noted the absence of emission factors for military waste. Their characterization can be used to increase the accuracy of these and future efforts, improve military sampling strategies in the future, and to reduce harmful exposure to military personnel. Additionally, emission factors that represent

both conditions, with and without plastics, can be incorporated into epidemiology studies to determine relationships between exposure and illnesses.

Preview

This thesis uses the scholarly article format. Chapter 2 contains the article resulting from the research, which will be submitted to the *Environmental Science and Technology (ES&T)* journal. The article provides the body of this thesis and contains all the elements of research in its layout as prescribed by the peer reviewed journal. As an independent chapter, it includes an abstract, introduction, materials/methods, and results/discussion sections. A supplementary information file will be submitted with the article and included in the appendices of this thesis. Chapter 3 offers a final discussion of the article conclusions along with pertinent findings and future research not discussed in Chapter 2.

II. Scholarly Article

Submitted to *Environmental Science and Technology* journal 2012 (www.pubs.acs.org)

Emissions from Simulated Open Burning of Deployed US Military Waste

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Abstract

U.S. military forces have relied primarily on open burning as an expedient method of volume reduction and treatment of solid waste during the conflicts in Afghanistan and Iraq. This study is the first effort to characterize a broad range of pollutants and their emission factors during the burning of military waste and the effects that recycling efforts, namely removing plastics, might have on emissions. Piles of simulated military waste were constructed, burned, and emissions sampled at the U.S. Environmental Protection Agency (EPA) Open Burn Testing Facility (OBTF), Research Triangle Park, NC. Three tests contained polyethylene terephthalate (PET #1 or PET) plastic water bottles and four did not. Emission factors for polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), particulate matter (PM_{10}, $PM_{2.5}$), polychlorinated and polybrominated dioxins/furans (PCDD/F and PBDD/F), and criteria pollutants were determined and are contained within. The average PCDD/F emission factors were 270 ng-toxic equivalency (TEQ) per kg carbon burned (ng-TEQ/kg Cb), ranging from 35-780 ng-TEQ/kg Cb. Limited testing suggests that targeted removal of plastic water bottles has no apparent effect on reducing pollutants and may even promote increased emissions.

Introduction

Disposal of US military solid waste at deployed locations has traditionally been a logistical challenge and failure to dispose of such waste can reduce personnel combat effectiveness through exposure to pathogens and vector borne illnesses. The lack of sufficient and safe off-base disposal methods in the deployed environment, combined with limited numbers of waste disposal devices such as incinerators, have forced continued reliance on open burning in "burn pits" as an expedient method of volume reduction and treatment for solid waste during the conflicts in Afghanistan and Iraq. As one of the most extreme examples, records show that civilian contractors at Joint Base Balad, Iraq, one of the largest military bases and central logistics hub in Iraq at the time, were burning nearly 200 short tons per day of solid waste during peak troop levels in 2008 (Armed Forces Health Surveillance Center: Naval Health Research Center & US Army Public Health Command, 2010; Belasco, 2009).

Medical studies of military personnel returning from the conflicts in Afghanistan and Iraq have reported significant increases in respiratory illnesses (Smith et al., 2009; Szema et al., 2011). The emissions from open burning of military waste have been implicated as a potential cause (Szema et al., 2011). In response, the U.S. Department of Defense (DOD) has published guidance regulating burn pit operations, including limiting their use and prohibiting the burning of recyclable plastics (Trimble, 2010). Also, the health service components of the U.S. Army, Navy, and Air Force began coordinating efforts to develop better waste disposal and exposure monitoring methods. Additionally, the Department of Veterans Affairs commissioned the Institute of Medicine to assess the long-term health effects from exposure to burn pits in Afghanistan and Iraq, and their

findings were released in 2011 (Tollerud et al., 2011). These efforts have all expressed the need to characterize emissions in order to assess potential exposure hazards from open burning of solid waste at deployed locations. This is accomplished through the determination of emission factors for use in combination with a waste burn activity level to determine exposure. To date, no effort has been made to determine emission factors for deployed military waste.

Emissions from anthropogenic fuel sources, such as burning of domestic waste, have become increasingly scrutinized as a significant global source of harmful pollutants. The most commonly investigated of these harmful pollutants are polychlorinated dibenzodioxins/dibenzofurans (PCDD/Fs). These pollutants are highly persistent and bioaccumulating, and the more toxic forms have been labeled as a "known human carcinogen" (Gullett et al., 2000). Research on emissions from open burning of solid waste at dumps (Barakat, 2003; Gullett et al., 2010) and burning of representative household waste in controlled laboratories (Gullett et al., 2007; Gullett et al., 2001; Hedman et al., 2005; Lemieux et al., 2000; Wyrzykowska et al., 2009; Zhang et al., 2011) has been conducted. Results from these studies show that 1-2 orders of magnitude variation in emission factors exists for these toxic compounds, likely due to burn conditions, waste composition, and orientation (Lemieux et al., 2000). Lemieux et al. compared the emission factors for burning domestic waste in burn barrels for avid recyclers and non-recyclers and determined that PCDD/F emissions were higher for avid recyclers than for non-recyclers (Lemieux et al., 2000). Their research concluded that combustion conditions, such as temperature profiles and oxygen availability, as well as the particular mixture of carbon and chlorine in the presence of a metal catalyst, which

are important factors in the formation of PCDD/Fs, were different between the two conditions and may be a possible explanation (Lemieux et al., 2000). In addition to these harmful pollutants, it is well known that copious amounts of other classes of pollutants are emitted from open burning and are also hazardous to both the environment and human health.

The study described herein provides a characterization of the broadest range of health-related pollutants from open burning conducted to date. Additionally, this study is the first attempt to characterize emissions from the open burning of simulated deployed military waste and the effects that current military recycling efforts, namely removing plastic water bottles, might have on emissions.

Materials and Methods

Waste Composition

Burn samples were constructed using a representative military waste composition, based on expert knowledge from DOD personnel (Hardt, 2011). Notable observed differences in military waste streams from that of civilian waste include large quantities of packaging, construction materials, food waste, canvas material, Meal Ready to Eat (MRE) waste, and plastic water bottles. Each component consisted of at least five pieces to help ensure even distribution throughout the sample. A total of seven tests were performed and each test used a burn sample weighing approximately 10 kg. Table 1 provides the composition of the burn samples by percent mass per category of waste. The complete burn sample descriptions are provided in the supporting information. Three tests were conducted with PET #1 water bottles included. Also, PET #1 plastic

was removed for four additional tests, to determine if targeted recycling efforts aided in reducing harmful burn emissions. With PET#1 bottles removed, the mass of the remaining components were increased proportionally to achieve the 10 kg sample size as reflected in Table 1.

Table 1: Composition of Burn Samples

Waste Category	Waste Description		Target Mass %
Plastics	PET-#1 (4.5, 0) HDPE-#2 (0.5, 0.52) PP-#5 (1.5, 1.57) PVC-#3 (1, 1.05)	PS-#6 (1.5, 1.57) PU (foams)-#7 (0.5, 0.52) ABS (electronics) (0.5, 0.52)	10
Wood	Treated (pallets) (3, 3.14) Untreated (3, 3.14)		6
Metals	Aluminum/Tin (1.4, 1.47) Iron/Steel (1, 1.05) Copper Wire (w/ Insulation) (1, 1.05)	Magnesium (0.1, 0.1) MRE Heaters (0.25, 0.25) Paint Can (0.25, 0.25)	4
Misc. Comb.	Fabrics (synthetic) (3, 3.14) Fabrics (natural) (6, 6.28) Canvas (military) (2, 2.09) Neoprene (0.1, 0.1) Nylon (0.1, 0.1) Cardboard (6, 6.28) Paper (18, 18.85) Rubber (2.5, 2.62)	Food (18, 18.85) Wet Food (slop) (13, 13.61) Oils and Grease (2, 2.09) Unopened MREs (1, 1.05) Opened MRE Inner Packaging (2.8, 2.93) Soap (0.1, 0.105) Batteries (0.1, 0.105) Cigarette Waste (0.3, 0.31)	75
Dunnage	Glass (3, 3.14) Building Materials (2, 2.09)		5
		TOTAL	100

(#,#) = (% mass with plastics, % mass without plastics) (Hardt, 2011)

Experimental Setup

Small-scale emissions testing on this simulated deployed military waste was conducted at the US Environmental Protection Agency (EPA) Open Burning Testing Facility (OBTF), Research Triangle Park, North Carolina. This facility has been used in previous EPA emission studies (Gullett et al., 2007; Gullett et al., 2001; Gullett et al.,

2003; Lemieux et al., 2000; Wasson et al., 2005). The OBTF consists of an enclosed metal burn shed with sheetrock covering the interior walls and ceiling and concrete blocks covering the plywood flooring. Additionally, the interior of the OBTF was covered with certified, ASTM B479, clean aluminum foil to ensure no contamination from previous experiments. A recirculation fan was placed inside the shed to ensure complete mixing. Outside air was supplied through a baffled vent at a rate of approximately one volume change per minute. Both the fan and the air vent were positioned to ensure no direct flow to the burning material, which would have affected burning conditions.

For testing, waste was piled randomly and loosely-packed onto a burn platform, located in the center of the OBTF. Food slop was then added to represent deployed dining facility waste. Under the burn platform was an electronic scale, which was tared, to monitor burn progression through mass loss. Samples were ignited in multiple areas using a propane torch. Once ignited, the OBTF was closed and emission monitoring began.

All emissions were extracted through a 25.5 cm transfer duct, throughout which flow rate and temperature were monitored. Thermocouples were placed under and above the burn pile and at the transfer duct. After sampling, emissions were ultimately sent to an incinerator, bag-house, and scrubber for pollution control purposes.

Sampling and Analysis.

Health-related compounds were the major compounds of concern for this research and included PXDD/PXDF (X = Cl or Br), polycyclic aromatic hydrocarbons (PAHs), volatile organic compound (VOCs), PM_{10}, $PM_{2.5}$, and criteria pollutants, including carbon

monoxide (CO), carbon dioxide (CO_2), sulfur dioxide (SO_2), various forms of nitric oxides (NOx), and oxygen (O_2). Table 2 provides an overview of the EPA method used for the sampling and analysis of each class of compounds.

Air sampling for PXDD/PXDFs was conducted inside the OBTF utilizing a Tisch Environmental TE-PNY1123 high volume sampler operating at >400 liters per minute. The flue gases were passed through a pre-cleaned quartz filter and sorbed onto a polyurethane foam (PUF) cartridge. The PUF was then analyzed using High Resolution Gas Chromatography with High Resolution Mass Spectrometry (HRGC/HRMS) on a Hewlett-Packard gas chromatograph 6890 Series equipped with a CTC Analytics Combi PAL auto-sampler (CTC Analytics, Switzerland) and coupled with a Micromass Premiere (Waters Inc., U.K.) double-focusing high-resolution mass spectrometer (U.S. Environmental Protection Agency, 1999b).

PAHs were sampled isokinetically from the transfer duct. Samples were withdrawn and collected in a multi-component sampling train which included a quartz-fiber pre-filter and a packed bed of porous polymeric adsorbent resin. Samples were analyzed with a Gas Chromatograph/Mass Spectrometer (GC/MS) for identification and determination of concentrations of compounds.

VOCs were sampled from the transfer duct utilizing a stainless steel vacuum SUMMA® canister. Sampling began upon burn initiation and was taken for approximately 45 minutes, which would bias these samples toward the flaming rather than smoldering phase of the burn. All samples were then analyzed via GC/MS (U.S. Environmental Protection Agency, 1999c). VOC sampling was conducted in accordance with EPA Method TO-14 (U.S. Environmental Protection Agency, 1999c).

Particulate matter was sampled at two locations. Model SP-280 air sampling pumps (Air Diagnostics and Engineering Inc., USA) operating at approximately 20 liters per minute were connected to PM_{10} and $PM_{2.5}$ 37-mm tared Teflon filters located inside the OBTF. These air sampling pumps were calibrated before and after each test using a Sierra Model 82251-M-1 calibrator. Gravimetric methods were used to analyze these samples. Again, these samples were biased to the flaming phase of the burn, as samples were collected within the first hour prior to possible clogging. Additionally, PM samples were taken from the transfer duct utilizing EPA Method 5 (U.S. Environmental Protection Agency, 2000). Once gravimetric analysis was completed, samples were analyzed for metals via Inductively Coupled Plasma-Mass Spectrometry (ICP-MS) methods.

Criteria pollutants and other key OBTF parameters were monitored using a continuous environmental monitoring (CEM) system. Sample gases were extracted through stainless steel probes located inside the burn shed at the inlet of the transfer duct and transported via Teflon® tubing where it passed through a Hankison dryer and particulate filter. The sample was then split, with a portion of the flow passed through a Drierte canister for additional drying before going to the individual CO, CO_2, and O_2 analyzers. The analog outputs of the analyzers were connected to a data acquisition system for monitoring and recording. Sampling frequency was set to occur at two second intervals. Each monitoring system was calibrated using a pre- and post-test three point calibration. A Horiba Model VIA510 was used to monitor CO concentrations and CO_2 concentrations higher than 1000 ppm, while a Fuji Model 3300A was used to monitor lower concentrations of CO_2. A continuous gas sample was also extracted from the

transfer duct for analysis with an Advance Pollution Instrumentation Model 100AH SO_2 analyzer for the determination of SO_2 concentrations (Advanced Pollution Instruments Inc., 1997). The Teledyne Model 200E NOx analyzer with an external Teledyne Model 501x thermal converter was used to determine NOx concentrations from a continuous gas sample (Advanced Pollution Instruments Inc., 1997). Finally, the Rosemont Model 755 analyzer was used to continuously monitor O_2 concentrations. Emissions were monitored for a minimum of three hours or until the CO_2 and waste bed temperatures neared ambient conditions.

Table 2: EPA Sampling and Analysis Method.

Class	EPA Method	Analysis Method
PXDD/ PXDF	TO-9A (U.S. Environmental Protection Agency, 1999b)	HRGC /HRMS
PAHs	0010 (U.S. Environmental Protection Agency, 1986) / 8270 (U.S. Environmental Protection Agency, 1996c)	GC/MS
VOCs	TO-14 (U.S. Environmental Protection Agency, 1999c) / TO-15 (U.S. Environmental Protection Agency, 1999a)	GC/MS
SVOCs	0010 (U.S. Environmental Protection Agency, 1986)	GC/MS
$PM_{2.5}$	05 (U.S. Environmental Protection Agency, 2000)	Gravimetric & ICP-MS (metals)
PM_{10}	05 (U.S. Environmental Protection Agency, 2000)	Gravimetric
O_2, CO_2	3A (U.S. Environmental Protection Agency, 1989)	CEM
CO	10A (U.S. Environmental Protection Agency, 1996a)	CEM
SO_2	6C (U.S. Environmental Protection Agency, 1996b)	CEM
NOx	7E (U.S. Environmental Protection Agency, 1990)	CEM

Results and Discussion

Table 3 contains the general sampling data for the seven samples tested, three with PET#1 and four without. Sampling times ranged from 180 to 189 minutes. The percent of mass consumed for the duration of sampling, approximately 180 minutes, ranged from 35.7 to 66.1% with an average of 54.0%. Of the total mass consumed over the 24-hour period, an average of 83% occurred during the first three hours, which encompassed the progression from the flaming to smoldering stage for each sample.

Table 3: General Sample Information

Sample I.D.	Containing PET #1 (yes/no)	Duration (min)	Mass Initial (kg)	Duration Mass Final (kg)	Duration Mass Loss (kg)	24-hr Mass Final (kg)	24-hr Mass Loss (kg)
yp-afit-061311	Yes	180	9.51	6.11	3.4	-	-
yp-afit-061411	Yes	180	9.84	4.92	4.92	3.23	6.61
np-afit-061511	No	180	9.89	4.59	5.3	3.69	6.2
np-afit-061611	No	184	9.96	4.65*	5.5	2.69	7.27
yp-afit-061711	Yes	189	9.58	4.33	5.25	-	-
np-afit-062011	No	184	9.86	3.34	6.52	3.00	6.86
np-afit-062111	No	186	10.27	3.9	6.37	-	-

Sample naming convention = yp/np (yes/no plastics)-afit (Air Force Institute of Technology)-date (MMDDYY)
* = No final mass datum was gathered. Therefore, the average rate of mass loss for the other six samples was applied to determine a reasonable mass loss for sample 4.
- = no data gathered

All emission factors were calculated in mass of pollutant per mass of carbon consumed using the carbon balance method. This work assumed that all carbon emissions are in the form of CO and CO_2 and that neglecting other forms of carbon

emissions would not significantly alter the emission factors. Using measurements from the CEM system, flue gas flow rates were converted to volume of gas sampled at standard temperature and pressure, from which the molecular weight of the flue gas was determined from the partial pressures of the constituent gases. Finally, the carbon fraction and partial pressure of CO and CO_2, and the total volume of flue gas sampled was used to determine the mass of carbon sampled over the duration of the run. Additionally, the 2005 World Health Organization (WHO) toxic equivalent factors (TEFs) (Van den Berg et al., 2006) were applied to PCDD/F factors for determination of the toxic equivalency quotient (TEQ) and for comparison to previously published emission factors for anthropogenic waste. TEFs for PCDD/F were applied to their PBDD/F counterparts based on limited published toxicities (Samara et al., 2009).

Once all emission factors were determined, an Analysis of Variance (ANOVA) was run, utilizing the statistical software JMP 9.0® (SAS Institute, Inc., Cary NC) for all samples containing plastics versus all samples without plastics to determine if there was a statistical difference between the two conditions. A confidence interval of 95% (p-value < 0.05) was the criteria for determining statistical difference. For all emission factors that were not statistically different, the two sets were combined to calculate averages, yielding a sample size of 7. Table 4 provides the p-value, average emission factor, and standard deviation for PAHs, PM, and VOCs sampled that were the highest contributors in their class or listed as one of the EPA's urban air toxics and found to be statistically similar. Only six pollutants were determined to be statistically different and are provided in Table 5 along with the p-value, average, and standard deviation for both conditions. For these conditions, sample sizes of 3 and 4 were used, containing PET #1 and not containing PET

#1, respectively. The inability to statistically differentiate between the emissions factors for the two conditions is explained by the large variance observed for each emission factor. Future research using an increased sample size might provide more certainty on statistical difference. All emission factors for each sample are provided in the supporting information.

Table 4: PAH, PM, and VOC Emission Factors

PAH (n=7)	Average (µg/g Cb)	Standard Deviation	ANOVA p-value
Naphthalene [1]	410	79	0.65
Phenanthrene	120	44	0.52
Acenaphthylene	89	32	0.72
Fluorene	40	10	0.62
Fluoranthene	30	114	0.84
Pyrene	25	8.3	0.82
Anthracene	21	9.3	0.98
Chrysene	18	9.5	0.73
Benzo(a)anthracene	11	6.5	0.99
PM (n=5)			
$PM_{2.5}$	43000	13600	0.38
PM_{10}	46000	12800	0.79
VOC (n=7)			
Benzene[1,2]	2000	470	0.12
Propene	1700	580	0.17
Acetone	1600	580	0.12
Vinyl Acetate[1]	1500	570	0.42
Acrolein[1,2]	1200	360	0.16
Toluene[1]	860	120	0.18
1,3-Butadiene[1,2]	540	110	0.057
2-Butanone	540	200	0.078
Naphthalene[1]	360	180	0.72
Chloromethane	220	91	0.74
Carbon Disulfide[1]	160	100	0.65
m,p-Xylenes[1]	150	70	0.093
Acetonitrile [1]	100	31	0.46
Acrylonitrile[1,2]	44	9.5	0.28
Carbon Tetrachloride[1,2]	1.5	3.9	0.29

[1] = EPA List of Hazardous Air Pollutants (U.S. Environmental Protection Agency, 2008).
[2] = EPA Urban Air Toxics (U.S. Environmental Protection Agency, 1999d)

Table 5: Statistically Different VOC Emission Factors

	Plastics (n=3)		No Plastics (n=4)		
	Average (µg/g Cb)	Standard Deviation	Average (µg/g Cb)	Standard Deviation	ANOVA p-value
VOC					
Styrene[1]	3500	320	4200	380	0.050
Ethylbenzene[1]	340	94	510	74	0.039
o-Xylene[1]	49	2.8	80	20	0.049
n-Propylbenzene	21	3.1	34	7.3	0.044
Cumene[1]	18	2.8	33	5.2	0.0062
Methylene Chloride[1,2]	8.3	0.05	5.3	1.2	0.0085

[1] = EPA List of Hazardous Air Pollutants (U.S. Environmental Protection Agency, 2008).
[2] = EPA Urban Air Toxics (U.S. Environmental Protection Agency, 1999d)

All PXDD/F emission factors were determined and compared for statistical differences, as previously discussed. The individual PXDD/F emission factors along with p-values, averages, and standard deviations are provided in the supporting information. Additionally, TEFs, which are also provided in the supporting information, were applied to all PCDD/F, and their related PBDD/F congeners, to determine TEQ emission factors for comparison with published material. Table 6 provides the p-value, average, and standard deviation for PCDD/F and PBDD/F TEQ emission factors. The three runs comprising plastics and four runs comprising no plastics are combined as the ANOVA showed no statistical distinctions between their emissions. The TEQ emission factors for chlorinated dioxins/furans are an order of magnitude greater than brominated dioxins/furans. All individual PXDD/F emission factors are provided in the supporting information.

As evident in related research, large variances ranging 1-2 orders of magnitude were observed for many congeners both within and between the sample conditions. The

reasons for this large variance are not well understood, but are thought to be dependent on waste composition and burn characteristics (Gullett et al., 2001).

Table 6: TEQ-PXDD/F Emission Factors

Sample size = 7	Average (ng-TEQ/kg Cb)	StDev	ANOVA p-value
WHO2005-TEQ-PCDD	200	240	0.095
WHO2005-TEQ-PCDF	68	54	0.10
WHO2005-TEQ-PCDD/PCDF	270	290	0.092
WHO2005-TEQ-PBDD	2.8	2.8	0.22
WHO2005-TEQ-PBDF	17	17	0.22
WHO2005-TEQ-PBDD/PBDF	20	19	0.20

Although only six VOC emission factors were found to be statistically different, the averages for each condition were calculated to determine if any visible trends occurred to explore the effects of plastic recycling efforts on emission factors. Figure 1 illustrates the comparative averages between plastics vs. no plastics for $PM_{2.5}$, PM_{10}, and the average sums of all PAHs and VOCs comprised of 16 and 75 constituents, respectively. Error bars on the graphs represent the maximum and minimum values observed. As seen in the graph, removing plastics produced higher amounts of VOCs and PM_{10}, but lower PAHs and $PM_{2.5}$ on average. However, given the similar values and ranges between the two conditions, the effects of removing plastics are inconclusive regarding these categories of pollutants.

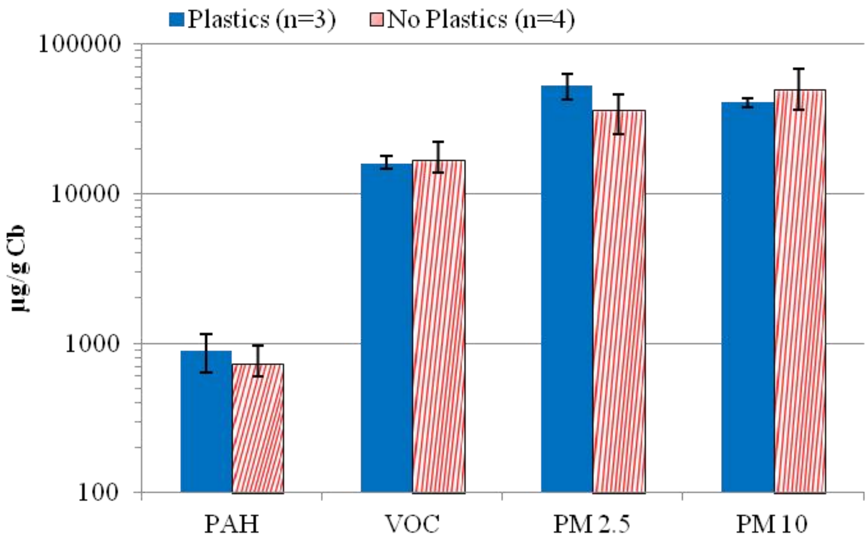

Figure 1: Average Total Emission Factors for PAH, VOC, and PM

Error bars depict the max and min observed values

Figure 2 illustrates the comparative averages for PCDD/F and PBDD/F TEQ emission factors along with the maximum and minimum observed values represented by error bars. For all PCDD/F and PBDD/F TEQ emission factors, the removal of plastics increased the emission factor and often produced a greater range of values. This increase is consistent with the burn barrel study by Lemieux et al. that compared the PCDD/F emission factors for non-recycling households to avid recycling households and is likely due to a change in burn characteristics brought on by the removal of plastics (Lemieux et al., 2000). Additional figures comparing all individual PXDD/F emission factors, in which the same trend is visible, are provided in the supporting information.

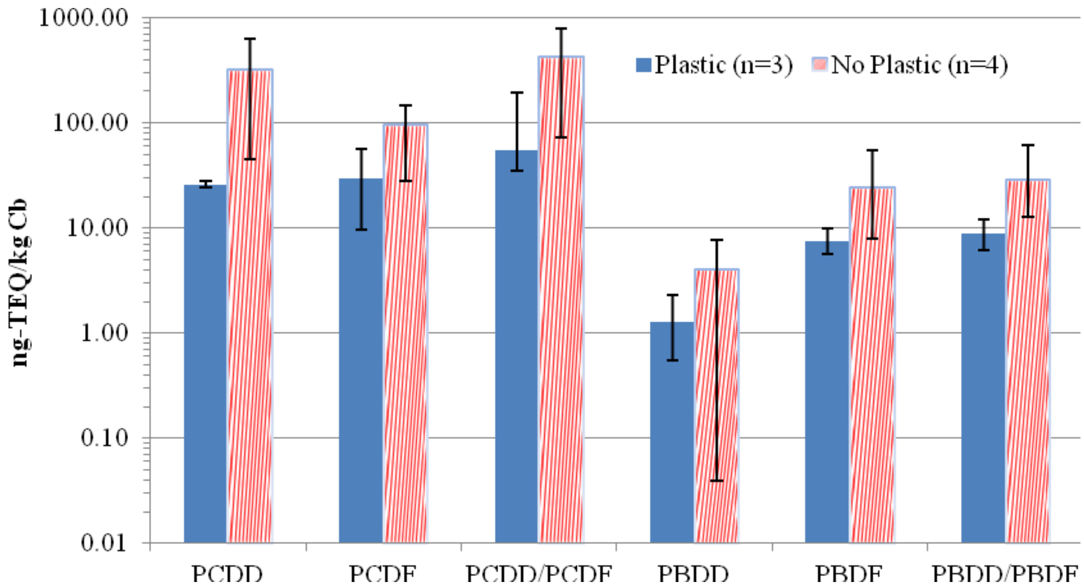

Figure 2: Average TEQ Emission Factors for PCDD/F and PBDD/F

The average total TEQ-PCDD/F emission factors for both conditions, samples containing plastics and samples not containing plastics, were compared to related emission studies. Both conditions were found to be within previously published TEQ-PCDD/F emission factors (Gullett et al., 2001; Hedman et al., 2005)**.**

The PCDD/F emission factors for simulated military waste were found to have a total average of 270 ng-TEQ/kg Cb; 55 ng-TEQ/kg Cb average for samples with plastics, and 420 ng-TEQ/kg Cb average for samples without plastics (Figure 2). The PCDD/F emission factors range from 35 to 780 ng-TEQ/kg Cb which falls within previously observed values. Additionally, though the amount of variance prevented the determination of statistical difference for emission factors between conditions, limited data show that the targeted removal of plastic water bottles has no apparent effect on reducing pollutants and may even result in increased production of PCDD/Fs and

PBDD/Fs pollutants. A possible reason for this is that PET contains no Cl and adds energy content to the waste, promoting more complete combustion. Therefore, the removal of PET plastic water bottles from the waste stream to improve emissions from open burning appears unsupported by this research.

This study provides valuable inputs for use in future research, which could directly benefit the health of military personnel, namely the refinement of military sampling and monitoring strategies. Additionally the emission factors determined herein will prove a valuable inclusion to current exposure assessment and epidemiology efforts.

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Supporting Information Available

Burn sample descriptions, individual emissions data for every sampled pollutant, and additional PXDD/F graphs comparing plastics vs. no plastic are provided as supporting information to the article and found in the appendices of the thesis. This material is available free of charge via the Internet at http://pubs.acs.org.

Bibliography

The references of this article are combined with the thesis following appendixes.

III. Conclusion

Chapter Overview

This chapter provides additional discussion on the research findings in relation to the original questions outlined in Chapter 1. The scholarly article being submitted to the *Environmental Science and Technology* journal contains all prominent results of the research. However, due to length constraints for the manuscript, the article does not include some of the results, which are located in the appendices of the thesis, and additional discussion of these results. This chapter provides a brief review of the findings and additional discussion of the results. It covers the limitations and the significance of the research, as well as future recommended research.

Review of Findings

The discussion below provides a review of the findings with regards to the research questions presented in the introduction and re-shown here as:

1) What are the emission factors from open burning of simulated deployed U.S. military waste?
2) What are the effects of polyethylene terephthalate plastic (PETE or PET#1) recycling efforts on these emission factors?

    2.1: Is recycling beneficial given the additional resource requirements?

1. Emission factors for simulated deployed military waste:

All emission factors were calculated in a mass of pollutant per mass of carbon burned. The full emission factor findings for all classes of pollutants investigated are

located in Appendix B along with averages and standard deviations. Additionally, the 2005 WHO-TEQ factors were applied to both PCDD/Fs and PBDD/Fs to compare these emissions with similar research efforts to determine emission factors for simulated domestic waste. It was determined that PCDD/F emission factors for simulated military waste were similar to that of related research efforts (Gullett et al., 2001; Hedman et al., 2005). As discussed in the article, large variances were observed for many of the congeners, which was evident also in related research efforts. It is believed that this variance is attributable to waste composition and burn characteristics. Regardless, this research is perhaps the largest study, to date, to characterize such a broad range of pollutants and their emission factors for military waste.

2. Effects of PET#1 recycling efforts on emission factors:

Seven piles of simulated military waste were burned and sampled, three with plastics and four with plastic water bottles removed, to determine the effects of recycling efforts on emission factors. An ANOVA was conducted for each emission factor to compare the two conditions using a confidence interval of 95% ($p < 0.05$) has the criteria for statistical difference. It was determined that only six pollutants, all VOCs, were statistically different. Therefore, the sample size used in determining the averages of each pollutant with no statistical difference was ($n=7$), while the six statistically different pollutant averages were calculated with $n=3$ and $n=4$ for samples with and without plastics, respectively. The inability to determine statistical differences for the emission factors is due largely to the small sample size and the amount of variance observed for each pollutant. An increase in sample size would likely enable the determination of statistical difference.

Although unable to determine statistical differences, averages for each pollutant were calculated for samples with and without plastics. These averages were compared to determine if any trends were visible. For all PCDD/F and PBDD/F congeners, the averages for samples without plastics were higher than samples that had plastics. Graphs illustrating this can be found in Appendix D. It is believed that the removal of plastics has no apparent effect on reducing pollutants and may even alter the burn characteristics in such a way as to promote the formation of these pollutants. Evidence for this alteration is provided in Appendix E, Figure 11, as temperatures for samples without plastics were higher than for samples with plastics. It should be noted that one temperature reading for samples containing plastics appears to be flawed as it remains at ambient temperature until the end, likely due to a faulty thermocouple.

Additionally, criteria pollutant concentrations such as CO, CO_2, SOx, and NOx were monitored with the continuous emission monitoring system. These data were used to create graphs that show the concentration of these pollutants over the duration of the test for comparison of samples with and without plastics. The graphs are provided in Appendix E. It was observed that emission concentrations for CO, SOx, and NOx achieved higher peak concentrations for samples without plastics, while CO_2 and O_2 appear to be similar for the two conditions. However, it should be noted that two readings for SOx for each condition appear to be flawed, as they remain at ambient levels for the duration of the test. Additionally, the reliance of any conclusions drawn for the bed temperature readings are questionable, as it was observed that burning activity was highly variable.

2.1: Is recycling beneficial given the additional resource requirements?

Although some benefit would be realized from the removal of plastics, in the form of waste reduction, it apparently has no effect on reducing pollutants and may even alter the burn characteristics in such a way to increase the emission of PCDD/Fs and PBDD/Fs. Therefore, any targeted removal of PET#1 plastic should be considered as a volume-reduction or recycling effort and not as a means to reduce potentially harmful emissions. A better use of these resources would be for the removal of PVC, which serves as a source of chlorine and has been shown to be correlated with increases in PCDD/F emission factors.

Limitations

As with any laboratory study, the ability to replicate actual conditions is nearly impossible. This is complicated further by the natural variations found in waste composition and burn characteristics. Additionally, samples were constructed based on expert estimations of military waste composition, using materials gathered from local sources. Therefore some discrepancies between samples and actual waste are bound to occur.

As previously mentioned, the small sample size hindered attempts to determine statistical differences between the two sample conditions. An increase in sample size could assist in reducing variances for each emission factor, which would increase accuracy and enable a definitive solution to the effects of plastic removal on emission factors.

Significance of Research

As discussed in Chapter 1, medical and epidemiology studies have expressed the need to determine the emission factors for the open burning of military waste in order to help determine the relationship of exposure to these emissions and the observed illnesses in returning military personnel. Additionally, the determination of these emission factors will assist in the improvement of military monitoring and sampling strategies, to better assess exposures to potentially harmful emissions while deployed.

Future Research

This study provides valuable inputs for use in future research, which could directly benefit the health of military personnel. The following is a list of future research opportunities:

- Repeat this research with an increase in sample size which would assist in reducing variance and determining statistical differences between the two conditions.
- Vary the amount of unusual items such as MRE heaters to determine their effects on emissions
- Refine military sampling and monitoring strategies utilizing these emission factors

The pursuit of these research opportunities would all provide beneficial insight and knowledge for application in the area of air pollution controls and risk reduction.

Appendix A. Simulated Deployed Military Waste Composition

Table 7: Military Waste Sample Composition with Plastics

	Waste	Mass per charge		
CATEGORY	% (*)	kg	g	ITEMS
PLASTICS	10	1	1000	
PETE - # 1	4.5	0.45	450	soft drink bottles, cleaner bottles
HDPE - #2	0.5	0.05	50	milk jugs, detergent bottles, fuel tanks for vehicles, storage sheds, plastic bags, water pipes, deodorant sticks, plastic film
PP - #5	1.5	0.15	150	flip-top bottles, insulation cables, cloth, folders, bags
PVC - #3	1	0.1	100	piping, inflatable structures, clothing (bags, jacket..), insulation electric wires , plastic film/wrap
PS - #6	1.5	0.15	150	fast food containers, yogurt container, disposable razors, packaging material (white balls), Styrofoam cups
PU (foams) - #7	0.5	0.05	50	mats, insulation foam, car seat foam, mattresses
ABS (electronics) - # 7)	0.5	0.05	50	automotive components
WOOD	6	0.6	600	
Treated (pallets)	3	0.3	300	pallets, wood desk/furniture
untreated	3	0.3	300	plank
METALS	4	0.4	400	
Aluminum/tin	1.4	0.14	140	aluminum cans, drum, metal cans
Iron/Steel	1	0.1	100	bed frame,
Copper Wire, Insulation	1	0.1	100	wire, electronic
Magnesium	0.1	0.01	10	Electronic devices, mobile phones
MRE Heaters	0.25	0.025	25	MREs
paint can	0.25	0.025	25	
MISC. COMBUSTIBLES	75	7.5	7500	
Fabrics, synthetic	3	0.3	300	army cots,(PET)
Fabrics, natural	6	0.6	600	army cots, army dresses
Canvas, military	2	0.2	200	canvas, bags
Leather	0	0	0	piece of leather
neoprene	0.1	0.01	10	water dress
nylon	0.1	0.01	10	tents, ropes, ponchos and military supplies
latex	0	0	0	medical use (gloves, masks..), bathing suits, water boots
lint	0	0	0	Band-Aid, dressing
Cardboard	6	0.6	600	cardboard box/packing
Paper	18	1.8	1800	papers, newspapers
Rubber	2.5	0.25	250	rubber, tire
Food	18	1.8	1800	fruit, meat, eggs, pasta….
Wet food waste (slop)	13	1.3	1300	Soup, creams,…
Oils and greases	2	0.2	200	oil, grease
Unopened MREs	1	0.1	100	MREs
Opened MRE Inner Packaging	2.8	0.28	280	
soap	0.1	0.01	10	soap
rock	0	0	0	pieces of rocks
batteries	0.1	0.01	10	batteries
cigarette waste	0.3	0.03	30	cigarette waste
DUNNAGE	5	0.5	500	
Glass	3	0.3	300	glass bottles
Building Materials	2	0.2	200	bricks, concrete, sand

Table 8: Military Waste Sample Composition without Plastics

	Waste	Mass per charge		
CATEGORY	% (*)	kg	g	ITEMS
PLASTICS	**5.75**	**1**	**575.9**	
PETE - # 1	0	0	0	soft drink bottles, cleaner bottles
HDPE - #2	0.52	0.0524	52.4	milk jugs, detergent bottles, fuel tanks for vehicles, storage sheds, plastic bags, water pipes, deodorant sticks, plastic film
PP - #5	1.57	0.157	157.1	flip-top bottles, insulation cables, cloth, folders, bags
PVC - #3	1.05	0.1045	104.7	piping, inflatable structures, clothing (bags, jacket..), insulation electric wires , plastic film/wrap
PS - #6	1.57	0.157	157.1	fast food containers, yogurt container, disposable razors, packaging material (white balls), styrofoam cups
PU (foams) - #7	0.52	0.0523	52.4	mats, insulation foam, car seat foam, mattresses
ABS (electronics) - # 7)	0.52	0.05	552.4	automotive components
WOOD	**6.28**	**0.628**	**628.7**	
Treated (pallets)	3.14	0.314	314.1	pallets, wood desk/furniture
untreated	3.14	0.314	314.1	plank
METALS	**4.19**	**0.419**	**418.9**	
Aluminium/tin	1.47	0.147	146.6	aluminum cans, drum, metal cans
Iron/Steel	1.05	0.105	104.7	bed frame,
Copper Wire, Insul.	1.05	0.105	104.7	wire, electronic
Magnesium	0.1	0.01	10	Electronic devices, mobile phones
MRE Heaters	0.25	0.025	25	MREs
paint can	0.25	0.025	25	
MISC. COMBUSTIBLES	**78.53**	**7.853**	**7853.4**	
Fabrics, synthetic	3.14	0.314	314.1	army cots,(PET)
Fabrics, natural	6.28	0.628	628.3	army cots, army dresses
Canvas, military	2.09	0.209	209.4	canvas, bags
Leather	0	0	0	piece of leather
neoprene	0.1	0.0105	10.5	water dress
nylon	0.1	0.0105	10.5	tents, ropes, ponchos and military supplies
latex	0	0	0	medical use (gloves, masks..), bathing suits, water boots
lint	0	0	0	Band-Aid, dressing
Cardboard	6.28	0.628	628.3	cardboard box/packing
Paper	18.85	1.885	1884.8	papers, newspapers
Rubber	2.62	0.262	261.8	rubber, tire
Food	18.85	1.885	1884.8	fruit, meat, eggs, pasta….
Wet food waste (slop)	13.61	1.361	1361.3	Soup, creams,…
Oils and greases	2.09	0.209	209.4	oil, grease
Unopened MREs	1.05	0.105	104.7	MREs
Opened MRE Inner Packaging	2.93	0.293	293.2	
soap	0.105	0.0105	10.5	soap
rock	0	0	0	pieces of rocks
batteries	0.105	0.0105	10.5	batteries
cigarette waste	0.31	0.031	31.4	cigarette waste
DUNNAGE	**5.24**	**0.524**	**523.5**	
Glass	3.14	0.314	**314.1**	glass bottles
Building Materials	2.09	0.209	**209.4**	bricks, concrete, sand

Appendix B. Full Emission Factors

Table 9: Full VOC Emission Factors (ug/g Cb)

VOC	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
Styrene[1]	3900	3200	4600	3800	3500	4400	4100	0.050*	4000	490
Benzene[1,2]	2900	2100	2100	1700	1900	1500	1700	0.12	2000	470
Propene	1100	1500	1700	3000	1400	1700	1600	0.17	1700	580
Acetone	1000	1400	1900	2800	1300	1400	1600	0.12	1600	580
Vinyl Acetate[1]	940	1700	1500	2700	1300	1300	1300	0.42	1500	570
Acrolein[1,2]	880	1100	1200	2000	970	1100	1200	0.16	1200	360
Toluene[1]	830	860	930	1100	680	830	830	0.18	860	120
1,3-Butadiene[1,2]	400	470	560	740	510	570	550	0.057	540	110
2-Butanone (MEK)	260	430	660	900	480	490	570	0.08	540	200
Ethylbenzene[1]	280	450	540	580	280	520	410	0.039*	440	120
Naphthalene[1]	610	16	470	320	340	360	370	0.72	360	180
Chloromethane	240	300	290	320	75	150	170	0.74	220	91
Carbon Disulfide[1]	240	200	93	330	97	43	89	0.65	160	100
m,p-Xylenes[1]	100	90	160	300	120	170	140	0.093	150	69
Acetonitrile[1]	83	110	110	170	90	83	89	0.46	100	31
n-Heptane	64	110	93	160	59	99	70	0.34	94	36
Ethanol	80	78	76	150	97	78	70	0.75	90	27
n-Octane	63	96	85	160	53	89	63	0.34	86	35
n-Hexane	56	100	85	160	64	83	56	0.47	86	35
o-Xylene[1]	51	50	85	110	46	68	61	0.049*	67	22
n-Nonane	30	43	49	98	29	43	36	0.25	47	24
1,2-Dichloropropane	54	0	0	160	0	57	39	0.34	45	59
Acrylonitrile[1,2]	28	43	59	51	46	41	41	0.28	44	9.5
3-Chloro-1-propene	48	0	0	110	0	63	52	0.24	38	41
2-Hexanone	33	30	40	120	35	0	0	0.82	37	41
Tetrahydrofuran (THF)	25	38	42	56	26	23	25	0.52	34	12
n-Propylbenzene	20	25	32	44	19	29	29	0.044*	28	8.5
1,2,4-Trimethylbenzene	18	11	32	53	21	27	26	0.077	27	13
Cumene[1]	16	21	41	33	17	30	29	0.0062*	27	9.1
Methyl Methacrylate[1]	19	50	41	18	25	19	13	0.47	26	14
1,3,5-Trimethylbenzene	22	8.9	20	51	17	28	28	0.13	25	13
Bromomethane	10	33	42	16	38	14	9.6	0.54	23	14
d-Limonene	6.2	0	15	18	25	83	7.7	0.39	22	28
1,4-Dioxane[1]	24	15	14	35	6.1	24	17	0.33	19	9.4
Chloroethane	25	16	10	15	15	29	10	0.65	17	7.1
Chlorobenzene[1]	45	9.4	12	16	7.5	8.9	10	0.43	16	13
Vinyl Chloride[1,2]	27	21	18	8.2	7.1	13	10	0.31	15	7.4
4-Ethyltoluene	10	7.8	18	24	13	17	12	0.07	15	5.4
4-Methyl-2-pentanone	10	15	9.3	20	9.7	14	12	0.49	13	3.8
2-Propanol	13	0	22	23	9.0	9.9	11	0.12	12	7.8
Methylene Chloride[1,2]	8.3	8.3	6.3	6.1	8.2	3.7	5.1	0.0085*	6.6	1.8
Benzyl Chloride[1]	24	0	9.3	0	2.9	4.9	3.6	0.53	6.4	8.4
Ethyl Acetate	0	0	0	0	0	0	14	0.29	2.0	5.3
Carbon Tetrachloride[1,2]	10	0	0	0	0	0	0	0.29	1.5	3.9
1,2-Dichloroethane	0	0	0	0	0	5.1	0	0.44	0.73	1.9
Tetrachloroethene	4.8	0	0	0	0	0	0	0.29	0.68	1.8
1,1-Dichloroethene	2.8	0	0	0	0	0	0	0.29	0.40	1.1

Sample name convention: (yp/np) =yes plastic/no plastic, afit =Air Force Ins. Tech, Date (mm/dd/yy)
Non Detects (ND) = 0
(-) = no p-value calculated
(*) = meets p-value criteria < .05 to determine statistical distinction
[1] = EPA List of Hazardous Air Pollutants (U.S. Environmental Protection Agency, 2008).
[2] = EPA Urban Air Toxics (U.S. Environmental Protection Agency, 1999d)

Table 10: Statistically Different VOC Emission Factors (ug/g Cb)

	Plastics (n=3)		No Plastics (n=4)	
	Average	Standard Deviation	Average	Standard Deviation
VOC				
Styrene [1]	3500	320	4200	380
Ethylbenzene[1]	340	94	510	74
o-Xylene[1]	49	2.8	80	20
n-Propylbenzene	21	3.1	34	7.3
Cumene[1]	18	2.8	33	5.2
Methylene Chloride[1,2]	8.3	0.05	5.3	1.2

[1] = EPA List of Hazardous Air Pollutants (U.S. Environmental Protection Agency, 2008).
[2] = EPA Urban Air Toxics (U.S. Environmental Protection Agency, 1999d)

Table 11: Full PAH Emission Factors (ug/g Cb)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
PAH										
Naphthalene [1]	520	330	480	330	340	440	450	0.65	2900	79
Acenaphthylene	150	70	110	67	64	81	80	0.72	630	32
Acenaphthene	5.7	5.2	9.6	6.7	4.5	14	8.0	0.064	54	3.2
Fluorene	55	35	46	37	24	48	38	0.62	280	10
Phenanthrene	210	100	120	79	90	140	91	0.52	840	44
Anthracene	35	15	22	13	15	33	17	0.98	150	9.3
Fluoranthene	49	23	26	18	23	51	21	0.84	210	14
Pyrene	32	20	24	18	20	40	19	0.82	170	8.3
Benzo(a)anthracene	19	7.6	11	4.9	6.4	21	7.0	0.99	77	6.5
Chrysene	32	15	16	7.9	12	30	12	0.73	120	9.5
Benzo(b)fluoranthene	12	5.8	7.3	4.3	5.2	14	4.8	0.96	53	3.9
Benzo(k)fluoranthene	10	4.7	6.8	3.7	3.9	12	3.8	0.96	45	3.3
Benzo(a)pyrene	7.5	3.5	6.2	3.7	3.9	12	4.3	0.57	40	3.0
Indeno(1,2,3-cd)pyrene	5.7	2.3	4.0	1.8	2.6	5.8	2.1	0.95	24	1.7
Dibenz(a,h)anthracene	2.8	1.2	1.1	0.61	0.64	2.3	0.54	0.62	9.2	0.90
Benzo(ghi)perylene	5.7	2.3	4.0	2.4	3.2	6.9	2.7	0.87	27	1.8

[1] = EPA List of Hazardous Air Pollutants (U.S. Environmental Protection Agency, 2008).

Table 12: Full PM Emission Factors (mg/g Cb)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	Average	StDev	ANOVA p-value
PM										
PM 2.5	-	62500	42500	46400	38500	24900	-	43000	13600	0.38
PM 10	-	43500	37900	68100	43500	36600	-	45900	12800	0.79

Values for flawed samples were discarded and are represented with (–)

Table 13: Full PM Emission Factors (mg/dscm)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
PM										
PM 2.5	-	91.4	111	104	109	74.4	-	0.38	98.0	15.2
PM 10	-	96.3	99.4	138	123	102	-	0.79	112	18.0

Values for flawed samples were discarded and are represented with (–)

Table 14: Full PCDD/F Congener Emission Factors (ng/kg Cb)

WHO-2005 TEF	**Congener**	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
1	2378-Cl4DD	4.5	5.0	58	11	6.5	250	140	0.14	67	92
1	12378-Cl5DD	19	22	190	33	19	380	220	0.077	130	140
0.1	123478-Cl6DD	0.87	2.0	9.3	1.7	0.82	25	18	0.096	8.2	9.7
0.1	123678-Cl6DD	1.3	4.9	10	2.7	1.5	30	29	0.12	11	13
0.1	123789-Cl6DD	1.1	3.7	9.5	2.4	1.4	25	24	0.10	9.6	11
0.01	1234678-Cl7DD	0.70	5.2	3.4	1.9	1.6	13	20	0.23	6.6	7.4
0.0003	OCDD	0.044	0.48	0.096	0.15	0.20	0.50	1.0	0.49	0.36	0.35
0.1	2378-Cl4DF	24	9.5	62	16	7.5	46	36	0.86	29	20
0.03	12378-Cl5DF	8.4	3.0	18	4.5	1.8	19	20	0.063	11	8.1
0.3	23478-Cl5DF	120	41	180	53	21	320	240	0.11	140	110
0.1	123478-Cl6DF	59	32	76	30	7.6	150	160	0.13	73	58
0.1	123678-Cl6DF	45	21	61	22	6.1	110	120	0.12	54	44
0.1	123789-Cl6DF	31	11	20	15	2.6	38	71	0.25	27	23
0.1	234678-Cl6DF	52	28	62	41	7.6	160	180	0.11	76	68
0.01	1234678-Cl7DF	12	12	12	9.1	1.4	33	55	0.20	19	18
0.01	1234789-Cl7DF	3.4	2.1	1.5	1.4	0.26	3.4	9.9	0.43	3.1	3.2
0.0003	OCDF	0.22	0.33	0.073	0.13	0.014	0.19	1.0	0.58	0.28	0.33

Table 15: Full TEQ-PCDD/F Emission Factors (ng-TEQ/ kg Cb)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
WHO2005-TEQ-PCDD	24	28	250	45	26	640	370	0.095	200	240
WHO2005-TEQ-PCDF	57	23	83	28	9.5	150	130	0.10	68	54
WHO2005-TEQ-PCDD/PCDF	81	51	340	74	35	780	500	0.092	270	290

Table 16: Full PCDD/F Homolog Emission Factors (ng/kg Cb)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
Homologs										
TCDDs	270	250	6400	740	420	16000	5900	0.12	4200	5700
PCDDs	190	250	2800	420	190	7000	4400	0.090	2200	2700
HxCDDs	210	580	2200	420	200	6400	5300	0.10	2200	2600
HpCDDs	160	1000	810	480	300	2800	3900	0.19	1300	1400
OCDD	150	1600	320	490	680	1700	3500	0.49	1200	1200
TCDFs	11000	3000	25000	5200	3900	25000	17000	0.10	13000	9500
PCDFs	6600	2100	12000	2900	1500	18000	14000	0.097	8200	6600
HxCDFs	4600	2100	5800	2500	630	12000	12000	0.12	5700	4800
HpCDFs	2400	2100	1900	1600	250	4900	9400	0.25	3200	3100
OCDF	720	1100	240	420	47	650	3300	0.56	930	1100

Table 17: Full PBDD/F Emission Factors (ng/kg Cb)

WHO-2005 TEF	**Congener**	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
1	2,3,7,8 - TBDD	0	0	2.9	2.0	0	0	0	0.22	0.70	1.2
1	1,2,3,7,8 - PeBDD	0.50	0.78	1.8	0.96	2.1	6.9	0	0.52	1.89	2.3
0.1	1,2,3,4,7,8 + 1,2,3,6,7,8 - HxBDD	0	0.61	1.2	0.80	1.3	5.0	0.32	0.41	1.3	1.7
0.1	1,2,3,7,8,9 - HxBDD	0	0.38	1.0	1.5	0.99	3.2	0	0.29	1.0	1.1
0.01	1,2,3,4,6,7,8 - HpBDD	4.1	3.3	3.1	3.3	3.3	10	0.61	0.77	4.0	2.9
0.0003	1,2,3,4,6,7,8,9 - OBDD	6.1	6.6	0	0	0	9.5	6.8	0.97	4.1	4.0
0.1	2,3,7,8 - TBDF	2.7	3.6	0	9.0	11	140	10	0.43	24	50
0.03	1,2,3,7,8 - PeBDF	2.7	3.4	6.2	13	5.8	0	11	0.37	6.0	4.6
0.3	2,3,4,7,8 - PeBDF	6.4	6.5	11	18	11	96	12	0.33	23	32
0.1	1,2,3,4,7,8 - HxBDF	23	30	32	92	40	86	83	0.054	55	30
0.01	1,2,3,4,6,7,8 - HpBDF	110	150	120	410	140	340	310	0.072	230	130
0.0003	1,2,3,4,6,7,8,9 - OBDF	81	93	67	74	60	100	100	0.57	83	17

Non Detects (ND) = 0

Table 18: Full TEQ-PBDD/F Emission Factors (ng-TEQ/ kg Cb)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	ANOVA p-value	Average	StDev
WHO2005-TEQ-PBDD	0.55	0.91	4.9	3.2	2.3	7.8	0.04	0.22	2.8	2.8
WHO2005-TEQ-PBDF	5.7	6.9	7.9	20	9.9	54	16	0.22	17	17
WHO2005-TEQ-PBDD/PBDF	6.2	7.8	13	23	12	62	16	0.20	20	19

Appendix C. Emission Factor Conversion

Conversion to per waste burned

$$\frac{\text{TEQ}}{\text{kg}_{\text{carbon}}} \text{ x } \frac{\text{Carbon Burned}}{\text{Mass of Waste Burned}} = \frac{\text{TEQ}}{\text{Wate Burned}} \qquad (1)$$

Conversion to per mass of initial waste

$$\frac{\text{TEQ}}{\text{Waste Burned}} \text{ x } \frac{\text{Mass of Waste Burned}}{\text{Mass of Waste Initial}} = \frac{\text{TEQ}}{\text{Mass of Waste Initial}} \qquad (2)$$

Where:
Oxidation Factor = Amount of carbon in the fuel oxidized (assumed to be 100%)
% Carbon in Fuel = (Assumed to be 50%)
Mass of Waste Burned = Mass of Waste Initial – Mass of Waste Finial
Carbon Burned = Mass Loss * Oxidation Factor * % Carbon in Fuel

Table 19: Full TEQ-PCDD/F per Waste Burned (ng-TEQ/ $kg_{waste\ burned}$)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	Average	StDev	Average W/ Plastics	Average W/out Plastics
WHO2005-TEQ-PCDD	12	14	130	23	13	320	590	160	220	13	260
WHO2005-TEQ-PCDF	28	11	41	14	4.8	74	65	34	27	15	49
WHO2005-TEQ-PCDD/PCDF	40	25	170	37	18	390	650	190	240	28	310

Table 20: Full TEQ-PCDD/F per Initial Mass of Waste (ng-TEQ/ $kg_{waste\ initial}$)

	yp-afit-061311	yp-afit-061411	np-afit-061511	np-afit-061611	yp-afit-061711	np-afit-062011	np-afit-062111	Average	StDev	Average W/ Plastics	Average W/out Plastics
WHO2005-TEQ-PCDD	4.3	7.1	68	12	7.1	210	110	60	78	6.2	100
WHO2005-TEQ-PCDF	10	5.7	22	7.8	2.6	49	40	20	18	6.1	30
WHO2005-TEQ-PCDD/PCDF	14	13	90	19.8	9.7	259	150	80	95	12	130

Appendix D. PXDD/F Comparison Graphs

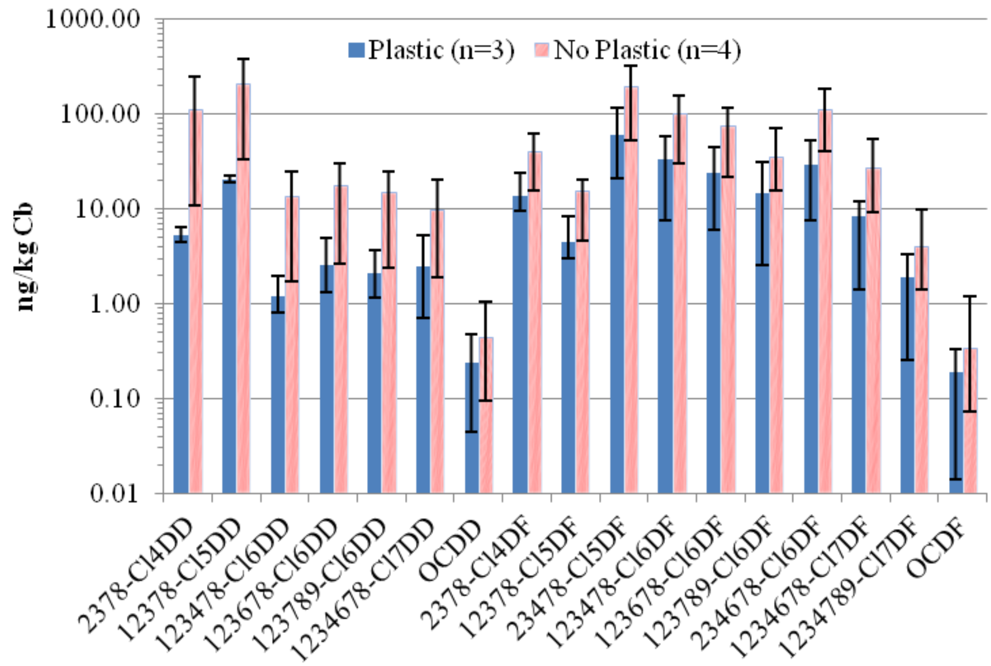

Figure 3: Average PCDD/F Congener Comparison Graph

Error Bars represent min and max values

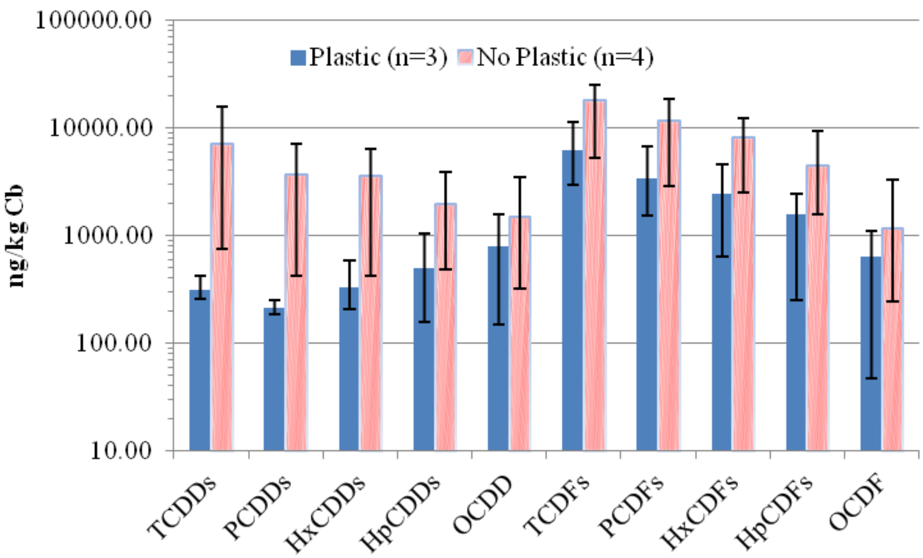

Figure 4: Average PCDD/F Homolog Comparison Graph

Error Bars represent min and max values

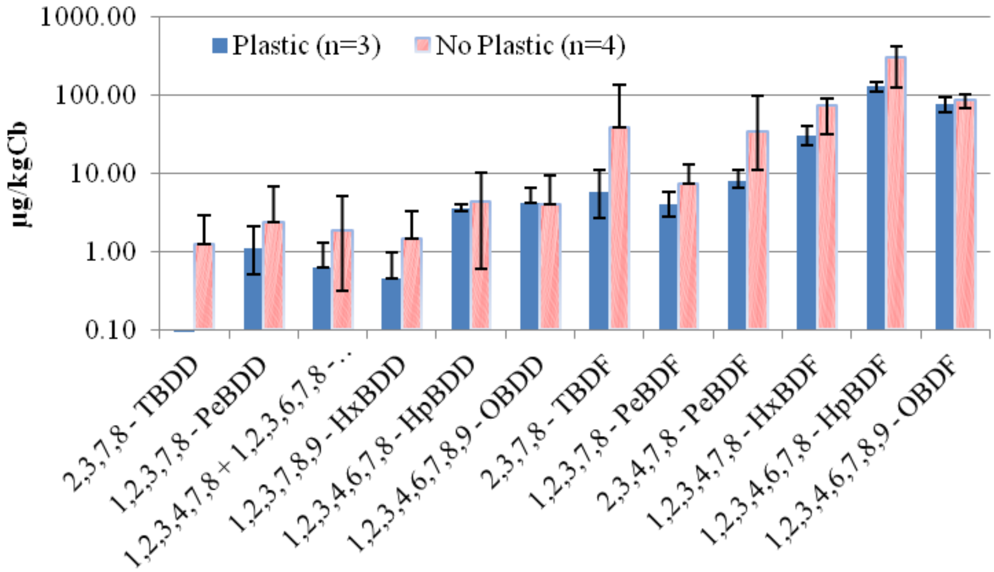

Figure 5: Average PBDD/F Congener Comparison Graph
Error Bars represent min and max values

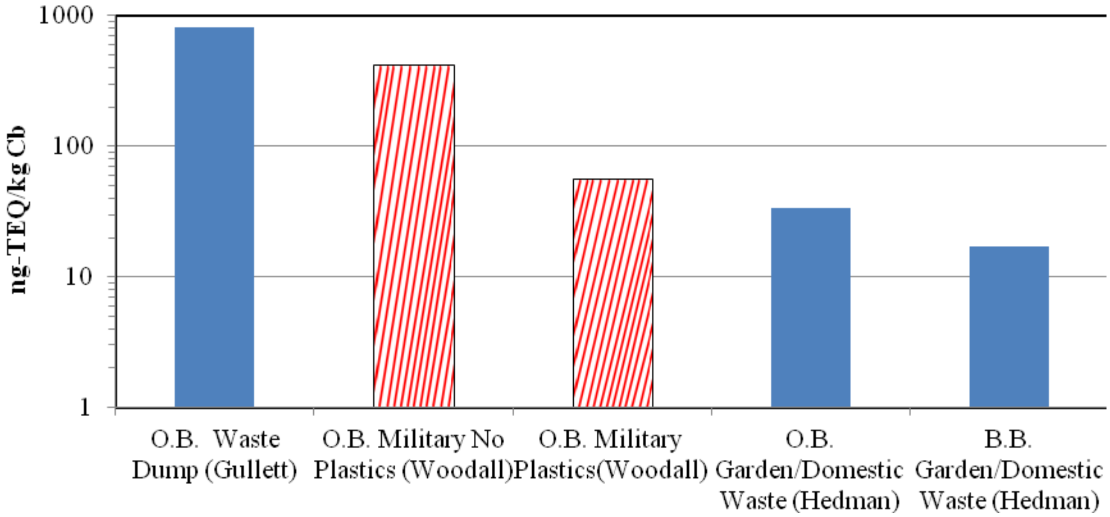

Figure 6: Total TEQ PCDD/F Comparison to Publish Values

(Gullett et al., 2001; Hedman et al., 2005).
B.B. = burn barrel, O.B. = open burning

Appendix E. Continuous Emission Monitoring Graphs

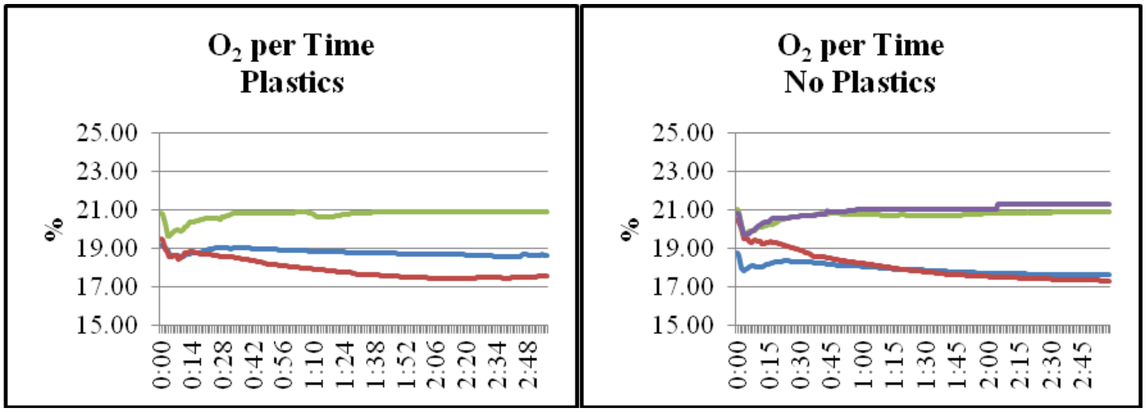

Figure 7: Comparison Graphs for O_2 per Time

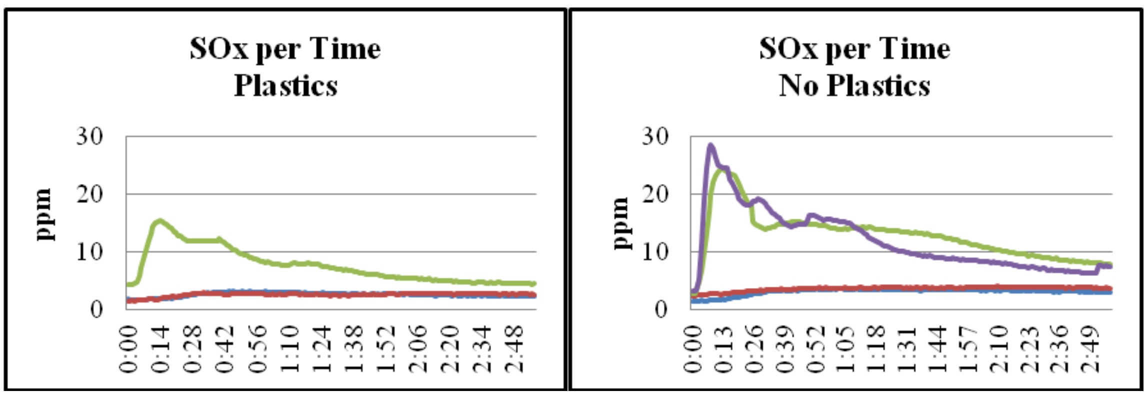

Figure 8: Comparison Graphs for SOx per Time

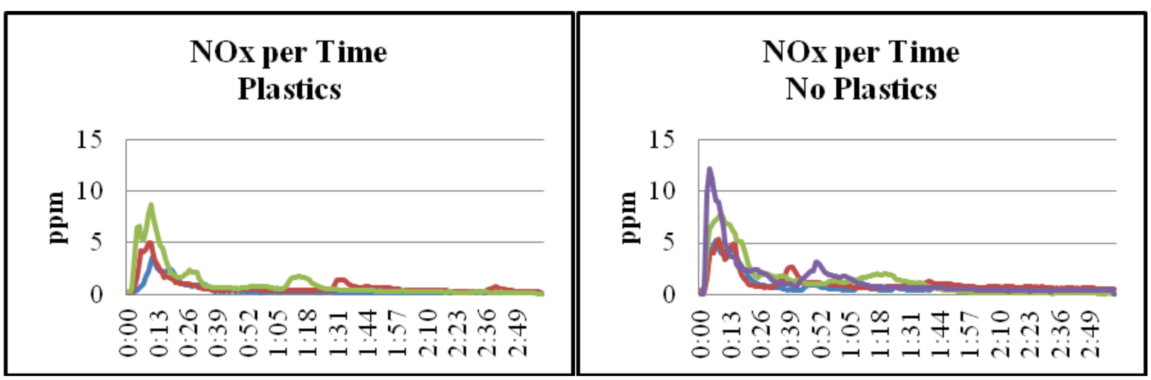

Figure 9: Comparison Graphs for NOx per Time

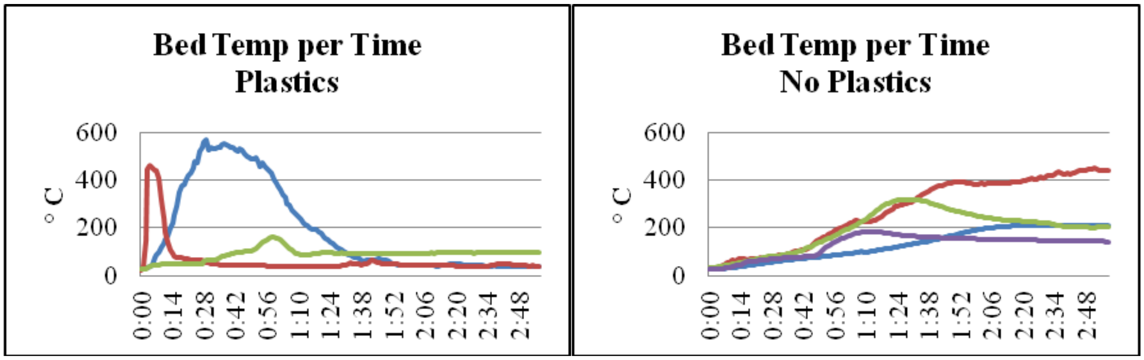

Figure 10: Graphs for Bed Temperature per Time

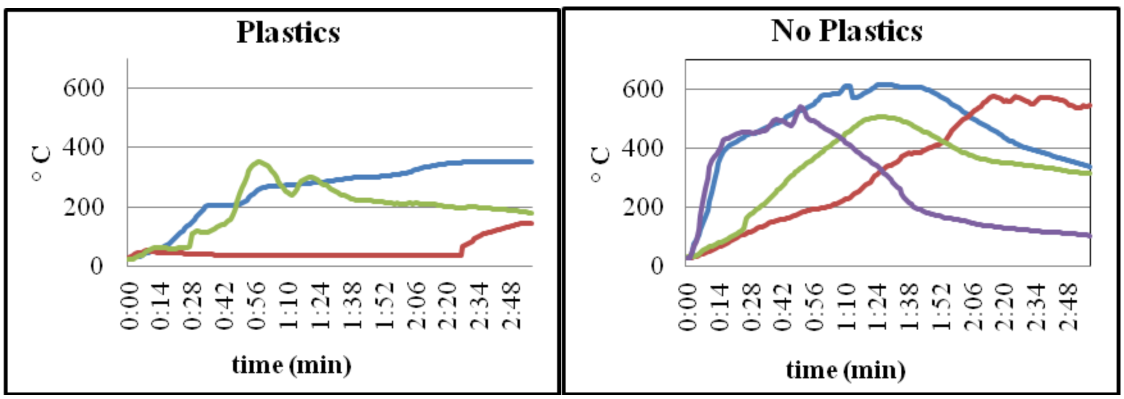

Figure 11: Comparison Graphs for OBTF Temperature per Time

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14. ABSTRACT
U.S. military forces have relied primarily on open burning as an expedient method of volume reduction and treatment of solid waste during the conflicts in Afghanistan and Iraq. This study is the first effort to characterize a broad range of pollutants and their emission factors during the burning of military waste and the effects that recycling efforts, namely removing plastics, might have on emissions. Piles of simulated military waste were constructed, burned, and emissions sampled at the U.S. Environmental Protection Agency (EPA) Open Burn Testing Facility (OBTF), Research Triangle Park, NC. Three tests contained polyethylene terephthalate (PET #1 or PET) plastic water bottles and four did not. Emission factors for polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), particulate matter (PM10, PM2.5), polychlorinated and polybrominated dioxins/furans (PCDD/F and PBDD/F), and criteria pollutants were determined and are contained within. The average PCDD/F emission factors were 270 ng-toxic equivalency (TEQ) per kg carbon burned (ng-TEQ/kg Cb), ranging from 35-780 ng-TEQ/kg Cb. Limited testing suggests that targeted removal of plastic water bottles has no apparent effect on reducing pollutants and may even promote increased emissions.

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